Ecological Engineering 114 (2018) 115–128

Contents lists available at ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Carbon sequestration in wetlands, from science to practice: An overview of the biogeochemical process, measurement methods, and policy framework

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ARTICLE INFO

Article history:
Received 25 December 2016
Received in revised form 16 June 2017
Accepted 17 June 2017
Available online 26 June 2017

Keywords:
Organic matter decomposition
Carbon sink capacity
Carbon flux
Global change factors
REDD+
Blue carbon

ABSTRACT

Ecosystem services are becoming increasingly important and a reason to promote the sustainable use of natural resources. Wetlands provide many valuable ecosystem services, including carbon (C) sequestration. Wetlands are an important C sink, playing a key role in climate regulation. As such, their ability to sequester C is being considered in national GHG emissions assessments and private initiatives as a potential source of revenue to manage carbon-balanced landscapes and pay for ecosystem services. To be able to implement these initiatives widely some aspects of wetland carbon science and practice still need to be formalized and standardized. Here we synthesize the scientific basis of the biogeochemical processes that drive C sequestration in wetlands and assess the methods available for its measurement. We have reviewed data in 110 peer-reviewed studies form wetlands around the world and provide an overview of the current policies and guidelines in which C sequestration in wetlands is framed as an ecosystem management practice. The intention of this review is to provide a wide and comprehensive summary of C sequestration in wetlands, from science to practice. This analysis can help inform practitioners and landscape managers in future considerations regarding project design and policy implementation, improve current climate mitigation schemes and payment for ecosystem services frameworks, and foster the worldwide implementation of wetland restoration, creation, and conservation projects for sustainable development and climate change mitigation and adaptation.

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1. Introduction

Since the Millennium Ecosystem Assessment (2001–2005) and The Economics of Ecosystems and Biodiversity Initiative (2008–2010) evaluated the consequences of ecosystem changes on human well-being, ecosystem services have been considered increasingly valuable and a reason to promote the preservation and sustainable use of natural resources. Before that, Costanza et al. (1997), Pimentel et al. (1997), Daily (1997), de Groot et al. (2002), and Patterson (2002), among others, analyzed the value of the services provided to society by several ecosystems using environmental valuation techniques and ecological pricing methods (Hansen, 2009; Ninan and Inoue, 2013). These studies pointed out the highly valuable benefits that ecosystems provide to humans, and the costs that replacing those services by anthropogenic means would entail. In addition, Day et al. (2009) emphasized the increasing importance of ecosystem services in supporting human economy as energy becomes scarcer and, consequently, the pressing need to manage ecosystems sustainably. About 10 years after the Millennium Ecosystem Assessment, countries agreed in the UN Sustainable Development Summit (2015) to adopt 17 Sustainable Development Goals to promote a global action plant for sustainable development through ecosystem conservation.

International and non-governmental organizations have produced guidelines to account for these services. However, defining these services and assigning them a tradeable value remains challenging (Vo et al., 2012; Muradian et al., 2013). This challenge exists because (1) a value for an ecosystem function is significantly more difficult to assign than a value for its structural attributes (Alongi, 2011); (2) the appropriate indicators or metrics to evaluate functions are as challenging to select as the appropriate methodologies to quantify them (Meyerson et al., 2005; Alongi, 2011; Robinson et al., 2012); (3) there are no sufficient data, nor is there a proper framework to analyze it (Ringold et al., 2013); (4) the value of the

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https://doi.org/10.1016/j.ecoleng.2017.06.037
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service depends on its scale and magnitude in each specific geographical location (Meyerson et al., 2005); and (5) uncertainties rise as one projects further into the future since many ecosystem services reflect non-linear processes (Barbier, 2009; Chisholm, 2010).

The determination of ecosystem services in wetlands can be further challenging, as these are dynamic ecosystems that have characteristics of both mature and immature ecosystems and thus the current state of a wetland may not reflect its future trends or conditions (Odum, 1969; Mitsch and Gosselink, 2015). Moreover, scarce adequate and up to date records of wetlands around the world make it difficult to track the changes in their quality and quantity (Zedler and Kercher, 2005). The loss and the degradation of wetlands imply that they are no longer able to perform their ecosystem functions, and the services they provide are thereby lost. This is not a small loss; despite of covering a small percentage of the land's surface (between 3.2%–9.7%; Melton et al., 2012), several studies report that wetlands are significant carbon sinks that store about one-third of the organic soil C pool of the world (Gorham, 1991; Lal, 2008), and one of the most valuable ecosystems in terms of the services they provide (Costanza et al., 1997; Millennium Ecosystem Assessment, 2005; de Groot et al., 2006).

C sequestration is the wetland service that has received most attention in recent years in the climate scenario, due to scientific evidence on the role that wetlands play in the radiative balance of earth's atmosphere (e.g. Roulet et al., 2007; Sjögersten et al., 2014), and to the emergence of policies introducing mechanisms to integrate C sequestration into climate change adaptation schemes and ecosystem conservation initiatives (Lovelock and McAllister, 2013; Aziz et al., 2016). However, the role that wetlands play as C sinks varies widely depending on the wetland hydrogeomorphic type and landscape position (e.g. Bernal and Mitsch, 2012; Bernal and Mitsch, 2013b). Furthermore, even though wetland ecosystems sequester C in their soil, their actual C sink capacity depends on the net balance of its C fluxes. Methane (CH_4) emissions particularly, hinder the ability of many freshwater wetlands to function as a net C sink (Bastviken et al., 2011). Coastal wetlands such as salt marshes and mangroves, on the other hand, are able to build a significant soil C pool with little- to no-CH_4 emissions (Poffenbarger et al., 2011), resulting in a consistent C sink. Fig. 1a gives an overview of the C pools of biomes around the world, based on global estimates (see Zedler and Kercher, 2005). The abundance of tropical forests and their high biomass production yields a high biome C stock. Wetlands, on the other hand, represent a vast underground C stock despite of their little acreage. Global estimates of wetland pools (tons C per ha) indicate that boreal peatlands (Gorham, 1991; Jobbágy and Jackson, 2000; Mitra et al., 2005) and coastal wetlands (Mcleod et al., 2011; Pendleton et al., 2012; Hopkinson et al., 2012) are large C sinks (Fig. 1b). The rest of wetland ecosystems (all temperate and tropical freshwater wetlands, as denoted in Fig. 1b) comprise a large C pool, but there are no accurate global estimates per wetland class within this category that can be compared to global C pool estimates of mangroves, saltmarshes, and boreal peatlands.

In this manuscript we present an overview of the biogeochemical process of C sequestration in wetlands (Section 2), an analysis the methods available to measure C sequestration and sediment accretion rates (Section 3), a review of published studies on carbon sequestration and sediment accretion in wetlands of the world (Section 4), a description of the policy framework that sets the ground for the inclusion of wetlands into management practices (Section 5), and an overview of remaining knowledge gaps and future directions (Section 6). The intention of this review is to connect science and practice, to help in overcoming the abovementioned challenges of accounting wetland carbon sequestration as an ecosystem service. Our approach to do so consists on synthetizing the scientific basis of the sequestration process and measurement methods valuable for policy as well as land managers and stakeholders' decision making, and describing current policies and guidelines that frame C sequestration within ecosystem management practices of wetland protection, restoration, and creation.

2. Carbon sequestration in wetlands: process overview

Carbon sequestration in wetlands refers to the carbon dioxide (CO_2) taken from the atmosphere that is transferred and accumulated into the wetlands' soil C pool as soil organic matter (SOM). As SOM accumulates its C is sequestered over time, as a result of the balance between C inputs and outputs. Inputs are constituted mainly by C contained in organic matter (OM) from senesced vegetation in and around the wetland (autochthonous), as well as C dissolved and suspended in inflowing waters and runoff (allochthonous). C outputs can consist of dissolved and suspended organic C in outflowing waters, but the main source are the inorganic forms emitted as CO_2 and CH_4 from the mineralization of OM during decomposition. While high C inputs can lead to high C accumulation, they can also yield high gas emissions. For example, Nahlik and Mitsch (2010) reported that CH_4 emissions were correlated with cumulative aboveground biomass productivity (ANPP) in the same wetlands that Bernal and Mitsch (2013a) found a strong correlation between ANPP and C sequestration in the soil. This indicates that the more C inputs to the soil, the more C to be potentially sequestered as SOM, but also the more abundant is the substrate to be degraded and exported to the atmosphere and to downstream waters (Freeman et al., 2004a). Therefore, C sequestration in wetlands cannot be understood without comprehending the factors that affect C decomposition and mineralization. These factors are described below.

- Substrate availability

The mechanisms that drive OM decomposition include a variety of biological, physical, and chemical processes that interact (Ewing et al., 2006; Baisden and Parfitt, 2007; Rasmussen et al., 2007; Schmidt et al., 2011), favoring the decomposition of easy-to-degrade organic materials by soil microbes (Laanbroek, 2010; Basiliko et al., 2012; de Graaff et al., 2014) and leaving behind to accumulate in the soil the recalcitrant components that microbes cannot further degrade efficiently (Davidson and Janssens, 2006; Bernal et al., 2016). Factors that are known to slow down OM decomposition and favor its accumulation in the soil are: scarcity of nutrients that limit microbial growth (Kuzyakov, 2002; Kuzyakov, 2010), physical protection or organic particles through aggregates (Six et al., 2002), and high content of organic compounds with low degradability (e.g. lignin; Hernes et al., 2007), among others. These physicochemical factors define the bioavailability of organic substrates and thus their recalcitrance.

The process of SOM decomposition is carried out by microbes through microbial growth and metabolism. These microbes obtain energy from the oxidation of organic compounds using electron acceptors in metabolic pathways that follow a predictable sequence based on their availability (Craft, 2001). To oxidize these organic compounds, microbes produce enzymes that aid in the degradation of OM (Kang et al., 2013); the more recalcitrant the compounds, the more necessary the enzymes to degrade them, and the higher the energetic cost for the microbial community to break down recalcitrant compounds into easier-to-degrade units. Microbially-available substrates become thereby scarce when SOM is recalcitrant. In addition to energy and electron acceptors, microbial growth requires nutrients that may not be supplied in enough

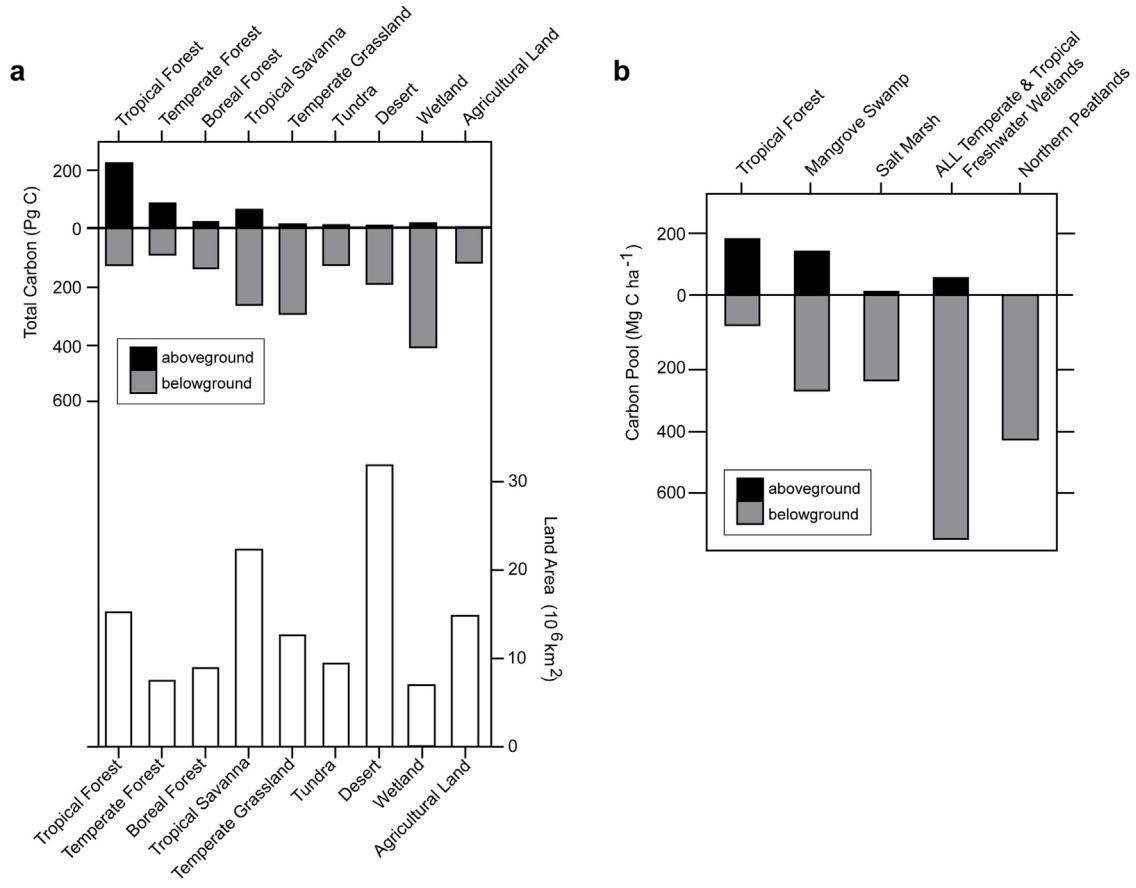

Fig. 1. (a) Estimated land area (10^6 km^2) and total carbon pool (1 Pg = 10^{15} g) in the top meter of some of the world's biomes, selected per their large carbon content. The biome 'wetland' includes all wetland types around the world. Figure modified from Zedler and Kercher (2005). (b) Estimated carbon pools (Mg C ha^{-1}; 1 Mg = 10^6 g) of the wetland biome, compared to the tropical forests. Tropical forest estimates from Brown and Lugo (1982); coastal wetland estimates (mangrove swamps and salt marshes, or intertidal forested wetlands and intertidal marshes in the Ramsar Classification system, respectively) from Mcleod et al. (2011), Pendleton et al. (2012), and Hopkinson et al. (2012); northern peatlands from Gorham (1991), Mitra et al. (2005), and Zedler and Kercher (2005). Note that the freshwater wetland category includes all freshwater wetlands in the world, covering all wetland types found in temperate and tropical climates; calculated by subtracting coastal wetland and northern peatland carbon stock and area from global estimates of wetland carbon stock and area (Gorham, 1991; Mitra et al., 2005; Zedler and Kercher, 2005).

abundance through the degradation of SOM, becoming "limiting nutrients" because, if they are lacking, OM cannot be mineralized nor incorporated in the microbial biomass (Hobbie, 2000; Fontaine et al., 2004; Hobbie, 2005).

- Temperature and oxygen

Decomposition of SOM, as any biological reaction, is expected to double every 10 °C increase (Davidson and Janssens, 2006). Temperature controls microbial metabolism and can shift microbial populations, affecting directly the SOM degradation process and the production of SOM-degrading enzymes (Razavi et al., 2016). Temperature also influences plant productivity, metabolism, and phenology (Baldwin et al., 2014), thereby affecting C inputs to the soil. Warming is therefore predicted to increase plant biomass globally, but also decrease global soil C pools (Hagerty et al., 2014). It is suspected that temperature, however, does not drive SOM decomposition as much as substrate availability does (Giardina and Ryan, 2000; Davidson and Janssens, 2006). Especially in wetland soils, where low O_2 availability under anaerobic conditions due to water saturation is the key to wetland's slow decomposition and high SOM accumulation.

Anaerobic conditions leave microbes with metabolic pathways less efficient than aerobic O_2 respiration, shaping microbial communities to the availability of the substrate to be respired (Smith et al., 2003; Whalen, 2005). Anaerobic conditions also limit the activity of enzymes involved in SOM degradation, mainly for two reasons (Freeman et al., 2004a; Freeman et al., 2004b; Toberman et al., 2008). First, they may need the presence of O_2. That is the case of phenol oxidases, one of the few enzymes able to degrade recalcitrant phenolic (organic) materials such as lignin and humic compounds. Second, the compounds that accumulate under anaerobic conditions can inhibit SOM-degrading enzyme activity. For example, the activity of hydrolases, involved in the decomposition of SOM, is restricted by the presence of the phenolic compounds that tend to accumulate in wetlands due to the lack of phenol oxidase activity.

- Global change scenarios

A slight change in the equilibrium of the wetland ecosystem such as a change in hydrological regime, could increase soil aeration, thereby increasing the activity of enzymes that decompose recalcitrant SOM with potential consequences such as those described in Freeman et al. (2001), where doubling phenol oxidase activity in a peatland doubled the CO_2 production (i.e. it increased by two-fold the mineralization of organic matter). Similarly, a rapid increase of SOM mineralization, can follow an input of fresh or labile C into the wetland, which typically occurs though an increase in belowground biomass or through new inflowing eutrophic waters loaded with dissolved organic matter (DOM; Hamer and Marschner, 2002). The increase in SOM mineralization

after new labile C inputs is known as "priming effect" (Cheng, 2009; Fischer et al., 2010; Kuzyakov, 2010).

Priming effect is frequent in the rhizosphere, as plants provide the soil microbial communities with electron donors (organic C) through root exudates, thereby stimulating microbial activity and, consequently, SOM decomposition (De Nobili et al., 2001; Fontaine et al., 2007; de Graaff et al., 2014; Bernal et al., 2016). The effect of SOM priming on soil C pools can be of significance when ecosystem changes lead to a deepening of the root profile or to a rapid increase in belowground organic inputs to the soil. A deepening of the root profile has been observed in North American wetlands with the invasion of *Phragmites australis* (Windham, 2001; Rooth et al., 2003; Bernal et al., 2016; Mozdzer et al., 2016), while the increase and deepening of belowground inputs can be expected in scenarios of stimulated plant growth, such as under elevated CO_2 (Stover et al., 2007; Langley et al., 2009; Day et al., 2009) or warming temperatures (Updegraff et al., 2001; Baldwin et al., 2014; Koven et al., 2015). These global change scenarios can severely decrease the ability of deep peat-forming wetlands to function as C sinks, especially boreal and high-altitude peatlands, which support little vegetation due to cold temperatures in their regions (Roulet et al., 2007; Buytaert et al., 2006; Tarnocai and Bockheim, 2011).

While nutrient-rich waters can fertilize and stimulate the growth of wetland plants (Nelson and Zavaleta, 2012; Caplan et al., 2015), studies reveal that nitrogen fertilization decreases root production of most wetland grasses and sedges, and thus has the potential to decrease belowground C inputs to SOM (Langley et al., 2009; Langley and Megonigal, 2010; White and Ringelberg, 1998). This decrease threatens marsh stability and the ability of coastal wetlands to keep up with sea level rise (Morris et al., 2002; Kirwan and Megonigal, 2013; Kirwan et al., 2016). There are studies reporting the collapse of wetlands that receive nutrient rich waters over time, due both to decreased root production of its vegetation, and to increased SOM mineralization through the alleviation of a nutrient limitation (Kuzyakov, 2002; Fontaine et al., 2004; Anisfeld and Hill, 2012; Deegan et al., 2012). This is attributed to a primed loss of soil C pool due mineralization, once microbes override the nutrient constraint on SOM decomposition (Hobbie, 2000, 2005).

- Carbon export:

DOM is often disregarded when studying C sequestration, since DOM is mobile and transient and the SOM sequestered is a long-term C stock. In wetlands, however, DOM can be an important pool and large quantities of DOM can enter deep soil layers, getting retained in the mineral wetland soil and potentially sequestered (Guggenberger and Kaiser, 2003; Kalbitz et al., 2003). Soil DOM pool can come from surface standing water or from root exudates released into the soil pore water (Freeman et al., 2004a). The contribution of DOM to the soil C pool depends on the degradability or recalcitrance of the organic matter dissolved, and some authors considered it the source of C for deep pools that remain unreached by plant roots (Harper and Tibbett, 2013). While mineral wetland soils have high but finite DOM sorption capacity, DOM sorption in organic rich wetland soils tends to be low (Guggenberger and Kaiser, 2003; Freeman et al., 2004a).

The ability of wetlands to build up biomass through the photosynthetic capture of atmospheric CO_2 and sequester it in the long-term buried as SOM allows these ecosystems to function as greenhouse gas (GHG) sinks, thereby reducing the radiative balance of the atmosphere. However, the net effect of sequestering C in wetland soils can be offset by the export of carbon in the form of CH_4 that is emitted as the result of anaerobic decomposition of SOM. This challenges significantly wetland management for climate change mitigation, since CH_4 has warming effect 28 times more potent than that of CO_2 (on a mass basis over a 100-year period; Myhre et al., 2013). Global simulations (Melton et al., 2012) and space borne measurements (Bloom et al., 2010) indicate that around 25% of the CH_4 emitted to the atmosphere is produced by wetlands.

CH_4 production in wetlands is the result of microbes using the last electron acceptor (CO_2) in the metabolic pathway and thus, its production is curtailed upon other acceptors being available. This is the case in tidal salt- and brackish-water wetlands (i.e. marshes and mangroves), where abundant sulfate from dissolved salts is the primary electron acceptor, making these ecosystems emitters of non-CH_4 decomposition gases (Poffenbarger et al., 2011). When sulfate levels are low, microbes utilize methanogenic pathways and thus produce CH_4. Additionally, Boye et al. (2017) also found that wetlands with sulfur were preventing the solubilization of C, thereby protecting SOM from degradation. As a result, saline and brackish wetlands are typically more efficient sequestering C than freshwater wetlands (Hopkinson et al., 2012; Pendleton et al., 2012; Nahlik and Fennessy, 2016), in addition to have the relative "climatic advantage" of being a net GHG sink due to the release of none to negligible CH_4. For this reason these coastal wetland ecosystems are being considered in current international climate conversations and might be included in climate change policies as a tool to abate climate change (see Section 5).

3. Measuring carbon sequestration in wetlands

Carbon sequestration in a wetland is usually determined as the product of bulk density (BD), soil C concentration (CC), and accretion rate (AR):

$$\text{CSequestration}\,[\text{M}\,\text{L}^{-2}\,\text{T}^{-1}] = \text{BD}\,[\text{M}\,\text{L}^{-3}] \times \text{CC}\,[\text{M}\,\text{M}^{-1}] \times \text{AR}\,[\text{LT}^{-1}] \quad (1)$$

Bulk density is determined dividing the dry weight of undisturbed soil cores with known dimensions (Grossman and Reinsch, 2002), while soil C concentration is measured by the wet oxidation or dry combustion methods (Pansu and Gautheyrou, 2006). There is a wide suit of options to measure accretion rates, but a standard methodology applicable in all different wetlands and conditions is currently lacking.

Depending on how soil accretion is measured, methods can be grouped into direct measurements and dating methods. Direct measurement of accretion requires the use of some kind of benchmark. For example, markers can be applied in the soil to create artificial horizons (i.e. marker horizons). After time, soil cores are extracted and the accretion of deposited material is measured in reference to this marker horizon. Different materials like clay or brick dust (Goodman et al., 2007), glitter (Richard, 1978), sand (French and Spencer, 1993), or feldspar (Cahoon and Turner, 1989; Cahoon, 1994) can be used to create the marker horizon. Among these, glitter and sand can be less prone to bioturbation and resuspension (Harter and Mitsch, 2003). Rare-earth elements (e.g. dysprosium and samarium) can also be used as markers (Knaus and Gent van, 1989), even though their use is not widespread. Another commonly used method that allows for a direct measurement of the vertical deposition using a benchmark is the sediment erosion table (SET). The SET provides a constant reference plane in space from which the relative height of the sediments over time can be measured by a set of adjustable pins (see Boumans and Day (1993) for a full description of the apparatus and use). Sediment accretion rates with this method can be measured with up to ±1.5 mm precision. When combined with artificial marker horizons, this method also allows for the determination of shallow compaction. It is also a conventional method to estimate land subsidence (i.e. vertical accretion minus elevation change) in coastal areas (Cahoon et al.,

1995; Cahoon and Lynch, 1997). The same principle of SET measurements operates with single sedimentation pins (e.g. Aust et al., 2012; Curado et al., 2012).

Dating methods can determine accretion rates by establishing an age to a specific depth increment within the soil profile. The most common methods to date soil profiles are based on radiometric readings, with typical radioisotopes used for dating being ^{137}Cs, ^{210}Pb, and ^{14}C. ^{137}Cs is a radionuclide produced during nuclear fission. Its widespread distribution in the environment started in the mid 1900′s as consequence of nuclear weapon testing, and peaked by the year 1964 (Ritchie and McHenry, 1990; Smith et al., 2000). The peak in activity is apparent in the soil profile, and thus the depth at which the peak is located correspond to the year of highest activity (i.e. 1964). The accretion rate is then calculated assuming a constant deposition over time, from the depth of the peak in the soil profile divided by the years passed since 1964. The method to establish accretion rates using ^{210}Pb is based on the underlying principle that unsupported ^{210}Pb from the decay of ^{222}Rn in the atmosphere is deposited at a constant rate in the wetland soil surface. Following, accumulation with complementary radioactive decay is assumed to create a vertical distribution with a characteristic decrease in its activity as a function of depth, from which a chronology can be inferred (MacKenzie et al., 2011). There are different models applicable to the ^{210}Pb activity data generated for the soil profile to calculate the accumulation rate. The constant rate of supply (CRC) and the constant initial concentration (CIC; Appleby and Oldfield, 1978; Oldfield and Appleby, 1984) are the two most frequently used. ^{14}C in turn is formed naturally in the atmosphere by cosmic ray action upon nitrogen mainly, although open-air nuclear testing during the mid-1900′s also added ^{14}C to this atmospheric pool (Turetsky et al., 2004). Once dates are assigned to the soil at a specific depth, usually the bottommost, accretion is calculated as the depth divided by this age.

There are other methods to date accreted deposits different to radiometric dating, for example the dendrogeomorphic technique proposed by Hupp and Morris (1990). This technique emerged to estimate accretion rates in freshwater forested wetlands. It is based in the principle that initial tree roots, upon germination, grow just below the surface, constituting a natural marker from which depth of accretion can be measured in mature trees. The accretion rate can be estimated by dividing the depth from the soil surface to the root collar by the age of the tree that is calculated using conventional dendrochronology techniques. Historical records, when applicable, are also used to date soil profiles. These refer to traceable events in the sediment records that allow assigning a date to a certain depth. Examples of historical records applied to accretion rate estimation in wetlands include sedimentary hiatuses in floodplains due channel diversion (Ye et al., 2015), wetland creation (Bernal and Mitsch, 2013a), fly ash deposition from early settlements (Tang et al., 2015), pollen accumulation from introduced species (Gascó et al., 2006), ash from volcano eruptions (Adame et al., 2015), or statistically significant changes in soil characteristics along the soil profile (e.g. soil bulk density, total phosphorus, δ^{13}C or δ^{15}N; Bhomia et al., 2015).

Besides determining C sequestration rates using Eq. (1), some studies have also used approaches based in Eddy covariance measurements of net ecosystem exchange (NEE; e.g. Jones and Humphries, 2002; Saunders et al., 2007). NEE is the balance between C uptake through gross primary production of wetland' autotrophs and the C losses through both autotrophic and heterotrophic respiration. This approach is based on gas exchange and does not account for C fluxes (C exported or imported) in water. Therefore, the use of Eddy covariance measurements must be coupled with mass balance approaches of C loss through water exports to specifically account for the C accumulation process in the soil (Waddington and Roulet, 2000; Billett et al., 2004; Roulet et al., 2007).

The preference and use of any of the methods available should be based on systems' dynamics (in terms of hydrological and ecological functioning), but is subjective to the scope of the research or project and to logistical or budgetary constraints. For example, direct measurements of the vertical deposition of peat or sediments are more frequently used to determine short-time rates because they are usually less expensive and tend to be more accurate than radiometric methods (Thomas and Ridd, 2004; Nolte et al., 2013). Direct measurements, however, require the installation or application of a benchmark at the beginning of the period for which the accretion rate is to be measured; the fact that they are labor intensive needs to be taken into consideration when assessing the need of site replications. Dating of deposited material, in contrast, does not require the establishment of an initial benchmark. Since historical rates are measured, practitioners and researchers can observe changes in accretion through time (e.g. Ruiz-Fernández et al., 2016; Cearreta et al., 2013; Kim and Rejmánková, 2002). This set of techniques, however, have the relative disadvantage of requiring additional processing in the laboratory and the use of expensive equipment to measure the radioactivity of soil isotopes. Also, several errors may arise when assigning age to the soil deposits due to sampling and preparation (e.g. Ritchie and McHenry, 1990; Goslar et al., 2005; MacKenzie et al., 1997; Olid et al., 2008; MacKenzie et al., 2011), interpolations and model assumptions inherent to the methods for the analysis of isotopes activity profiles (e.g. Clymo et al., 1998; Blaauw et al., 2004; Blaauw and Christen, 2005; Andersen, 2017) and, specifically for the case of ^{14}C, potential errors can appear associated to the presence of "old carbon" in the deposits and the need to calibrate ages to calendar years (Stuiver and Polach, 1977). To help overcome such errors, many studies use coupled radiometric dating techniques or contrast the results of one radioisotope dating with results from other accretion measurement method, to allow for a better interpretation (e.g. Craft, 2012; Bernal and Mitsch, 2013b; Villa and Mitsch, 2014a).

4. Data review

We reviewed 110 independent studies for accretion and carbon sequestration rates. The systematic searching methodology followed and the references of the papers used in this review is presented as Supplementary Material. Fig. 2 shows the use of different methods in wetland studies to determine accretion and C sequestration rates. Out of the 1037 independent measurements reported, soil dating was the most common method used (68%) followed by direct measurements (31.9%), while only 1 study featured the use of NEE (Fig. 2a). The most common employed radioisotope to date soils was the ^{137}Cs isotope (40.7%), followed by ^{210}Pb (27.1%) and ^{14}C (23.1%). Dendrogeomorphic technique and historical records, together, were used only in 8.5% of the studies to date soils. Regarding direct measurements, marker horizons were by far the most common method used (78.5%), followed by sedimentation pins (19.6%), plates (1.2%) and sediment traps (0.6%).

Grouping available data from wetlands according to the Ramsar classification system (Ramsar Convention Secretariat, 2013), we found that measurements were most common in coastal or marine wetlands than in inland wetlands (52.9% vs 47.1%, Fig. 2b). Intertidal salt marshes and forests were the most studied coastal wetlands (43.7% and 7.7%, respectively), while permanent freshwater marshes were the most studied inland wetlands, followed by non-forested peatlands and freshwater, tree-dominated wetlands (19.1%, 14.3% and 11.3%, respectively). The most used method in each of these wetlands was ^{137}Cs in intertidal marshes, marker horizons and ^{210}Pb in intertidal forested wetlands, marker horizons in permanent freshwater marshes, ^{14}C in non-forested peatlands,

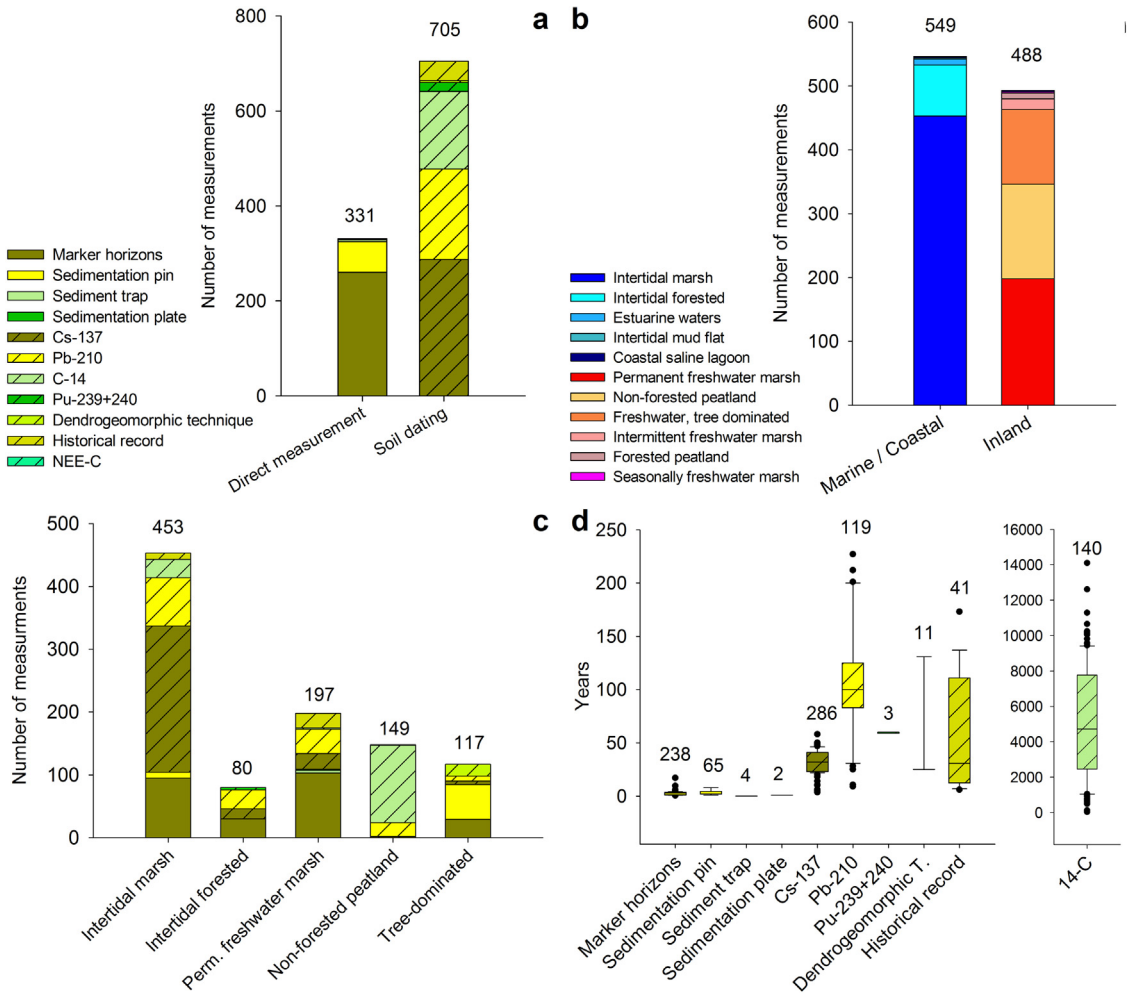

Fig. 2. Use of different methods to estimate sediment accretion and carbon sequestration rates in wetlands; a) Direct measurement and Soil dating determinations, showing the contribution of diverse methods to each; b) Number of measurements taken in the two major wetland types and wetlands ecosystems of the Ramsar classification system (i.e. marine/coastal and inland); c) Number of measurements under each of the different methods for the most commonly studied wetland classes; d) Boxplot of years for which accretion or carbon sequestration rates were calculated, showing the median (solid line in boxes) as well as the 25th (Q1) and 75th (Q3) quartiles as the edges of boxes.

and sedimentation pins in freshwater tree-dominated wetlands (Fig. 2c). Reviewed published data also indicates that the time for which accretion or C sequestration rates were measured varied considerable between methods. Direct measurements were performed for periods going from months to years (or few decades at the most), while soil dating allowed measurements covering periods ranging from decades to centuries (^{137}Cs and ^{210}Pb), up to thousands of years with ^{14}C (Fig. 2d).

The use of different methods to measure accretion rates in different wetlands and thus, in C sequestration rates, challenges the comparability between studies and muddles the gain of a general idea on its variability within and between wetlands. Reviewed data suggest that certain methods might be a better fit to certain class of wetlands, e.g. ^{137}Cs in intertidal marshes or ^{14}C in non-forested peatlands (Fig. 2c), while the dendrogeomorphic technique was used only in freshwater tree dominated wetlands (i.e. bottomland hardwood forests). In addition, there are methods more suited to be used across diverse wetland types. That is the case of marker horizons and ^{210}Pb, used with relative high frequency in the five most studied wetlands. The non-specificity of marker horizons and ^{210}Pb can be advantageous in landscape- or watershed-scale projects, because it avoids errors associated with the use of different measurement methods.

Regardless of the method used, the reviewed studies show that median accretion rates are statistically greater in permanent freshwater marshes, followed by freshwater tree-dominated wetlands, intertidal marshes, intertidal forested wetlands, and non-forested peatlands (Table 1, Fig. 3a, and Supplement Table S1). Median C sequestration rates in turn, were highly variable and statistically similar among these different wetlands except for the non-forested peatlands, which have a lower C sequestration rate than the rest of the wetland types (Table 1, Fig. 3b and Supplement Table S1). These non-forested peatlands present rates (geometric mean) around one fourth of that of the other wetland classes. The highest accretion rate (i.e. 172 mm yr^{-1}) was reported for a temperate created freshwater marsh in Ohio, USA, and was measured for a 0.8 yr period using marker horizons during (Mitsch et al., 2014a), whereas the highest C sequestration rate (i.e 1,784 g C m^{-2} yr^{-1}) was reported for a created salt marsh in a boreal climate, in which the sequestration rate was measured for a 32 yr period using historical records in the Yellow River Delta in China (Ye et al., 2015).

Out of the 333 reported carbon sequestration rates in this compilation, 40.5% were measured in boreal climates, 35.4% in temperate climates, 22.2% in the tropics, and 1.8% in arid climates (Fig. 4a). Sequestration rates measured in non-forested peatlands were the most commonly reported in boreal climates, while in temperate climates the most common measured wetland corresponded to intertidal marshes and forested wetlands. In tropical climates, in turn, the most common C sequestration estimates were taken in freshwater ecosystems. In arid climates, only rates in

Table 1
Soil accretion and carbon sequestration rates reported in 110 peer-reviewed studies. Rates are presented according to the Ramsar wetland classification system.

	Soil accretion (mm yr^{-1})				Carbon sequestration (g C m^{-2} yr^{-1})			
	Geometric mean	Median	Q1-Q3	n	Geometric mean	Median	Q1-Q3	n
Intertidal marsh	4.5	5.1	2.6–8.2	449	113.2	100	66.6–155	77
Intertidal forested	3.7	3.7	2.5–6.0	65	102.7	100	60–156.5	37
Permanent freshwater marsh	14.8	22.1	4.6–49	186	122.6	143	86.5–209.3	62
Non-forested peatland	0.7	0.5	0.3–1.3	88	26.1	20.0	15.1–38	130
Freshwater tree-dominated	9.2	12.0	5.0–20.4	117	176	186	89.5–563.3	14

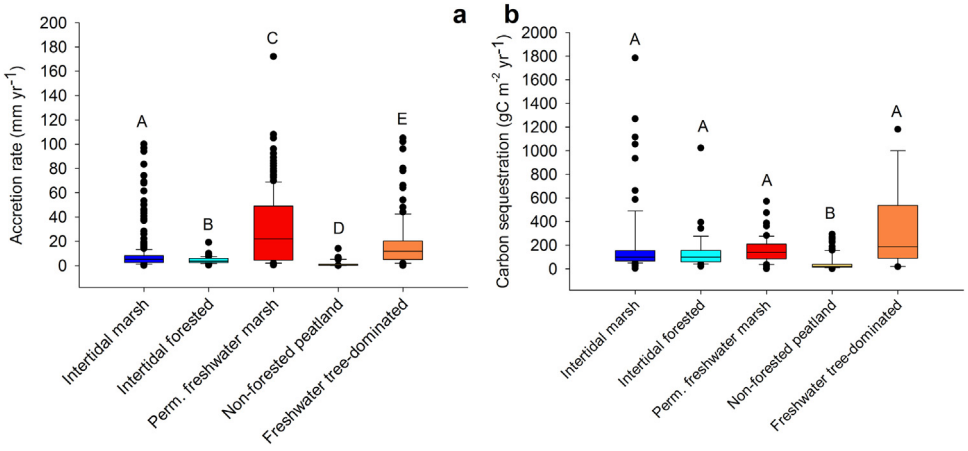

Fig. 3. Soil accretion and carbon sequestration rates reported in 110 peer-reviewed studies. a) Box plot of reported accretion rates. b) Boxplot of reported carbon sequestration rates for the most commonly studied wetland classes, grouped according to the Ramsar classification system. Horizontal lines in the boxes represent medians; upper and lower edges the 25th (Q1) and 75th (Q3) quartiles, respectively. Letters on top of boxplots indicate statistical difference according to the Mann-Whitney *U* test ($\alpha = 0.05$).

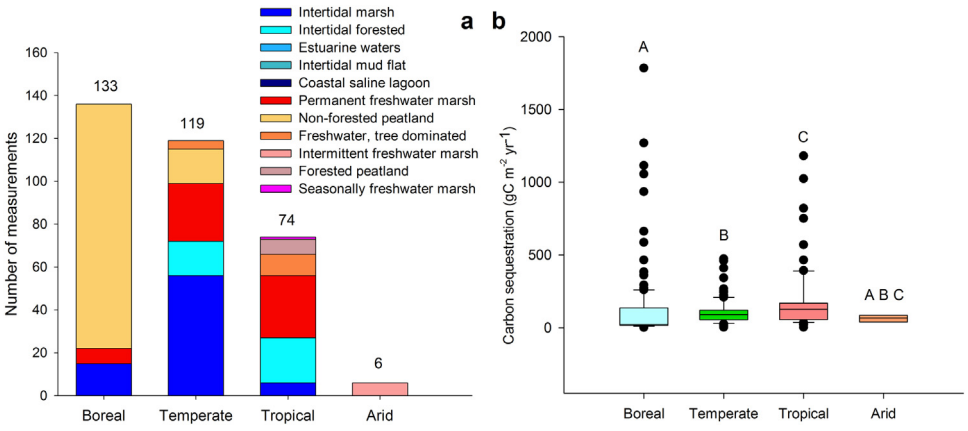

Fig. 4. Carbon sequestration rates along a latitudinal climate gradient, classified following the Köppen–Geiger Climate Classification System. Climates were assigned to each wetland site in the reviewed studies using descriptions in references or geographic coordinates on the map presented by Kottek et al. (2006). a) Number of C sequestration measurements per climate and wetland class. b) Boxplot of C sequestration rates reported in these peer-reviewed studies. Horizontal lines in boxes represent medians, and upper and lower edges the 25th (Q1) and 75th (Q3) quartiles, respectively. Letters on top of boxplots indicate statistical difference according to the Mann-Whitney *U* test ($\alpha = 0.05$).

intermittent freshwater marshes were measured. Median carbon sequestration rates per climate were greater in tropical wetlands (126.5 g C m^{-2} yr^{-1}), followed by temperate (89.8 g C m^{-2} yr^{-1}) and boreal wetlands (20.8 g C m^{-2} yr^{-1}; Fig. 4b, Supplement Table S2).

The geometric means of C sequestration vs. soil accretion rates for the five most studied wetland ecosystem classes fits a second order polynomial equation, indicating a Shelford-type relationship (Fig. 5). This parabolic limiting curve explains the importance of soil accretion in determining C sequestration rates across different wetlands, suggesting a higher C sequestration rates at intermediate rates of accretion. However, the explanation of the relationship between accretion and carbon sequestration rates observed in these wetlands surpass the simple description used by the Ramsar Classification System. Instead, they are better explained by a combination of vegetation and the underlying hydrogeomorphic setting (Brinson, 1993) that determines these different wetland ecosystems: Non-forested peatlands, most commonly known as bogs and mires, are predominantly feed through rainwater, exhibiting a non-sedimentary hydrogeomorphology. They are dominated by non-vascular vegetation with low productivity in cold climates (e.g. Tolonen and Turunen, 1996; Charman et al., 2015), thereby showing low accretion and C sequestration rates. Intertidal marshes and intertidal forested wetlands (i.e. salt marshes and mangroves, respectively) develop as riverine or fringe wetlands in coastal areas characterized by bi-directional horizontal water flows (tides). Tides represent an important subsidy, explaining the well-known high

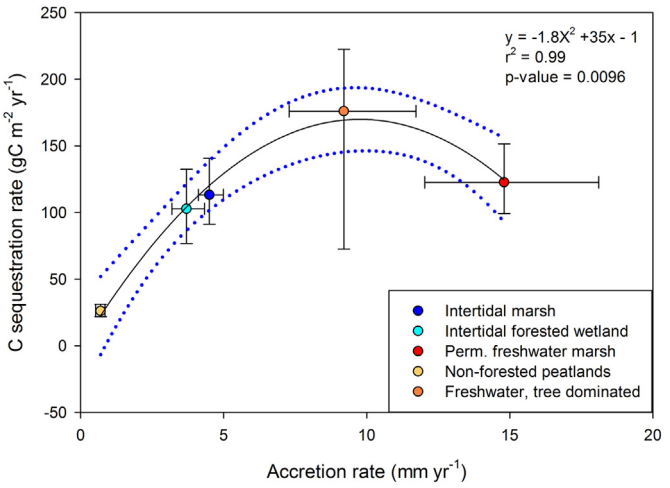

Fig. 5. Geometric means of the sediment accretion and C sequestration rates of the most commonly studied wetland classes, classified according to the Ramsar classification system. Error bars in each data point represent the asymmetric 95% confidence interval for the geometric mean; dotted lines represent the 95% confidence interval for a second-order polynomial regression line (in black).

productivity of intertidal wetlands. Despite of having substantially different biomass inputs to the soil (grasses in marshes and woody vegetation in mangroves), their sediment accretion and C sequestration rates are comparable and present similar variation, suggesting that the common constrain driving their sequestration rates are dominant predictable erosion-deposition cycles of their geomorphic setting (Adame et al., 2010; Rogers et al., 2013) rather than the source of organic matter accumulating in them (herbaceous vs. salt-tolerant trees). Freshwater, tree dominated wetlands (i.e. riparian forested wetlands and bottomland hardwood forests) develop in riverine settings with mostly inorganic deposits, but their horizontal water flows are uni-directional and intermittent, less frequent than those of intertidal wetlands. These are also subject to erosion and sediment loss (Zhang and Mitsch, 2007), but recalcitrant autochthonous OM accumulates in their soils (Chimner and Ewel, 2005; Bernal and Mitsch, 2012; Villa and Mitsch, 2014a), explaining their high and variable C sequestration rates. Permanent freshwater marshes, on the other hand, develop in riverine and depressional settings. Depressional settings tend to accumulate larger amounts of sediments than non-depressional ones do (Bernal and Mitsch, 2012), but their OM is not as recalcitrant as that of the tree dominated wetlands and most of the accumulated material is mineral (e.g. Ye et al., 2015).

While freshwater tree dominated have, on average, the highest C sequestration rates of the wetlands included in these published 110 studies, they are not the target of current mitigation policies due to their CH_4 emissions. These emissions are estimated to be higher than those from freshwater forested wetlands on organic soils (Sjögersten et al., 2014), and it is likely that emissions from freshwater forested wetlands have been widely underestimated as these estimates do not typically include important CH_4 emissions released from stems and canopies of trees (Pangala et al., 2013; Pangala et al., 2015; Terazawa et al., 2015).

5. Policy and practice frameworks

In the last couple of decades there has been an increasing emphasis on understanding and measuring C sequestration, that comes partially from the need to find ways to increase C pools in biomass and soils to mitigate the effects of GHGs in the atmosphere. Using ecosystems that naturally capture and sequester C is one of the most efficient and cost-effective approaches to counteract GHG emissions (Bedard-Haughn et al., 2006; Stern, 2007; Lal, 2008; IPCC, 2013). Alternative options like geological carbon capture and storage technologies are expensive and usually need energy inputs to function, while ecosystems that naturally sequester C imply no- or low additional cost. Additionally, a healthy ecosystem that sequesters C usually provides other valuable ecosystem services (Lal, 2010; Baig et al., 2016). Soils and oceans are the major C pools in the biosphere, with 2,500 and 38,400 Pg C, respectively (Schrag, 2007; Lal, 2008). However, these C sinks are decreasing their efficiency in absorbing anthropogenic emissions, as evidenced by a long-term increasing trend on the airborne CO_2 fraction (Canadell et al., 2007). Factors that decrease the C sink capacity of the land and oceans are deforestation, wetland loss, overexploitation of agricultural lands, and warmer ocean temperatures, among others. This, along with increasing anthropogenic GHG emissions, results in an altered C cycle that generates stronger and faster climate forcing than was initially anticipated (Canadell et al., 2007).

The decrease in the magnitude of C sinks is likely to decline even more under climate change and global population growth scenarios (Nellemann et al., 2009; Hopkinson et al., 2012), unless action is taken to preserve and enhance current C sinks. An efficient and multiscale enhancement of C sinks worldwide requires the implementation of a strong international policy framework and the provision of predictable and adequate financing mechanisms (Nhatumbo and Camargo, 2015), both of which are now part of the policies adopted by the United Nations Framework Convention on Climate Change (UNFCCC; Wylie et al., 2016). It is for these reasons that the agreement reached in Paris in the Conference of Parties in 2015 explicitly endorsed two important initiatives that had been in the works as pilot projects since the beginning of this century: (1) The nature-based initiative REDD+ (Reducing Emissions from Deforestation and forest Degradation), a framework that provides financial incentives to countries for reducing emissions associated with forest losses from conversion (deforestation) and degradation; and (2) the ecosystem-based adaptation to a changing climate strategy, to be achieved through the strengthening of a resilient development that recognizes the value of ecosystems and the co-benefits they provide, as well as their role complementing traditional actions such as infrastructure development (Baig et al., 2015, 2016).

a Policy frameworks

Countries under the UNFCCC prepare biennially their national GHG inventories in which they report up to date emissions and removals to the Convention. Those countries that are part of the Kyoto Protocol (i.e. developed countries listed in Annex I of the Convention) use the Intergovernmental Panel on Climate Change (IPCC) Guidelines to report on emissions and removals related to land use, land use change, and forestry (LULUCF; Iversen et al., 2014). The IPCC guidelines are the most comprehensive document on GHG reporting to the date (Climate Focus, 2011). The emissions trading set in Article 15 of the Kyoto Protocol established the mechanism through which countries could trade their “excess carbon” or “carbon credits”. These are emissions in CO_2 equivalents (CO_2e) allowed but not actually used. Similarly, Article 12 of the Protocol allows earning C credits with emission reduction projects in developing countries through the Clean Development Mechanism. For countries to be able to trade carbon credits from wetlands, wetland’s GHG fluxes and pools must be included in their national GHG inventory. This inclusion is now possible through the 2013 Wetlands Supplement to the 2006 Guidelines for National Greenhouse Gas Inventories (IPCC, 2014). The Supplement extends the content of the 2006 guidelines (IPCC, 2006) with updated information to reflect scientific advances (Iversen et al., 2014) and allows accounting for GHG emissions and reductions from drained inland

organic soils, rewetted organic soils, coastal wetlands, inland wetlands with mineral soils, and constructed wetlands for wastewater treatment. To the date, however, comprehensive spatiotemporal information on coastal wetlands C stocks and their drivers of C accumulation and emissions change has been limited around the world, and countries do not typically include them yet in the National GHG Inventories they submit to the Convention. Notwithstanding, the US has included them for the first time in the last Inventory (USEPA, 2017).

The IPCC guidance has been structured so that any country, regardless of experience or resources, can estimate emissions and removals with its national activity data at the simplest level using the provided default emission factors. The current default values provided for wetlands do not adequately cover wetland's sink capacity, because of scarce information on soil accumulation rates and on aboveground biomass production (Climate Focus, 2011). For example, the guidelines currently report default values for generic mangrove biomass production, despite of mangrove swamps (intertidal forested wetlands) being found as either shrubs or trees depending on latitude and species, thereby presenting high variability in their sink capacity when only one broad mangrove category is used. It is expected that these default values reported in upcoming editions of the guidelines will be more precise, as more data is available and financial mechanisms requiring accurate estimates are implemented.

Howard et al. (2017) point out that the development of a framework for climate mitigation policy is appropriate for ecosystems where (1) the C pool and current and potential fluxes are large enough to influence global C budgets and global net emissions, and (2) where human activity is the main driver increasing or decreasing the C pool and flux (i.e. through ecosystem degradation or restoration) and can be influenced with the implementation of a policy. The implementation of this policy therefore requires not only the understanding of the ecology and biogeochemistry of the ecosystem, but also identifying socioeconomic components such as land tenure and individuals or entities responsible for the ecosystem management. Coastal wetlands, being as large net C sink as forests, fit in this description. Available country information and monitoring on coastal wetlands, however, has not been as comprehensive as in forests (Climate Focus, 2011). This is changing thanks to the growing interest in "blue carbon" ecosystems (i.e. tidal marshes, mangroves, and sea grasses) as sustainable tools to mitigate and adapt to climate change.

The inclusion of wetlands as a sink or source of GHG is framed within the guidance of the Kyoto Protocol for LULUCF, which considers only wetlands on managed land where human intervention (such as water table modifications) changed its sink or source of GHG conditions (Iversen et al., 2014). This includes the category of "wetlands remaining wetlands" but, from a management perspective, the guidelines have set the attention on wetland creation and restoration alternatives (Roulet, 2000; Robertson, 2004; Dunn and Freeman, 2011), since these can be activities that generate revenue through C sequestered and thus GHGs reduced when a wetland is restored (Joosten, 2010; USEPA, 2017). This, however, may favor certain restoration projects based on the potential profitability selling carbon offsets. As Hansen (2009) pointed out, restoration of marshes (i.e. grassed wetlands) can have costs associated with restoration that exceed up to 87% the market value of C sequestration rates, while forested wetlands would have a higher economic potential with very few number of cases in which the costs of restoration exceed the market value of C.

b Prospects of financing mechanisms

The recent development of markets and payments for ecosystem services may incorporate private buyers in the funding schemes of wetland conservation (Engel et al., 2008; Gómez-Baggethun et al., 2010), opening the possibility for the inclusion of C sequestration as a service in pristine wetlands. These schemes differ from the Kyoto Protocol, as they may be based on a conservation principle of the current land-use and can be designed to include more than one ecosystem service (e.g. biodiversity, water supply, and C sequestration; Asquith et al., 2008; Pagiola, 2008; Wunder and Albán, 2008; Wendland et al., 2010). In practice, projects targeting exclusively wetlands are yet to be implemented but, given their economic value (Costanza et al., 1997; Woodward and Wui, 2001; Costanza et al., 2014), an increasing interest in integrating these ecosystems in projects that involve the payment for ecosystem services is expected, once a market is created (Mcleod et al., 2011; Pendleton et al., 2012; Silver et al., 2015).

In parallel, recent REDD+ initiatives financed through public and private sources started to include mangroves restoration and afforestation as GHG reduction strategy in existing policy frameworks (Howard et al., 2014; Sutton-Grier and Moore, 2016; Wylie et al., 2016). Mangroves have the advantage to fit better than most wetland ecosystems in current emission reduction frameworks such as REDD+, given that mangroves are forest ecosystems and that salinity of their coastal soils (and thus sulfur availability) impedes the CH_4 production that in freshwater wetlands compromises net GHG emissions (Gleason et al., 2009). The fact that coastal wetlands produce none to relative small quantities of CH_4 might accelerate their integration in GHG inventories (see USEPA, 2017) and the improvement of emission factors for wetland ecosystems in the IPCC guidelines. However, emission reduction frameworks (REDD+) are generally limited to biomass-C (estimated through biomass expansion factor, allometric equation, or aerial imagery methods; see VCS, 2010), and do not include yet the possibility of trading credits from the blue carbon stored in the soil, consequence as well of the limited guidance available to the date on wetland soil-C emission factors (Wylie et al., 2016). These frameworks, however, can incorporate and account for the numerous co-benefits that wetlands provide (Lovelock and McAllister, 2013; Nhatumbo and Camargo, 2015).

To include freshwater wetlands in emission reduction frameworks, their CH_4 emissions need to be accounted for (see Guidelines for National Greenhouse Inventories, REDD+ protocols, and private funded initiatives such as VCS, 2010). This approach involves the use of the global warming potential (GWP), the metric suggested by the IPCC to allow the trade-off between different forcing agents (i.e. GHGs; Forster et al., 2007). However, the following limitations must be taken into account when using GWP to account for CH_4 from wetlands: First, GWP is valid to compare discrete emissions, not those sustained through time (as is the case for CH_4 in wetlands). This issue was addressed by Neubauer and Megonigal (2015), who proposed as alternative metrics the sustained-flux global warming potential (SGWP, for emissions) and the sustained-flux cooling potential (SGCP, for uptake). Second, since GWPs are time dependent (e.g. 100-yr period), it is implicit in its use that emissions are assumed to remain constant during that period. However, CH_4 emissions in wetlands are known to change seasonally in response to temperature and hydrological regime (Bloom et al., 2010; Villa and Mitsch, 2014b). Lastly, the GWP value itself increases as the concentration of GHGs in atmosphere increases, and thus it has varied progressively from 21 to 25 to 28 since it was introduced in 1990 (Shine et al., 1990). These are not trivial limitations, and design of plausible financing mechanism targeting freshwater wetlands will need to address them all.

c Policy application and practice: wetland restoration for sustainable climate mitigation and adaptation

As described above, coastal wetlands fit current emission reduction policy frameworks. However, coastal wetlands have been rapidly disappearing due to human activities. Polidoro et al. (2010), estimated that the loss of mangroves is as fast as 8% of its annual coverage per year in SE Asia, with an average annual loss rate of 1% Kauffman et al. (2017). The main anthropogenic threats to coastal wetlands are both hydrological alterations (i.e. levees, impoundments, or upstream dams, among others), and the drainage of wetlands for their conversion to agriculture, livestock pastures, or aquaculture ponds (Climate Focus, 2011; Weston, 2014; Ahmed and Glaser (2016); Kauffman et al., 2017; Lovelock et al. (2017). Hydrological alterations often result in wetland subsidence and its concomitant disappearance into open water, which leads into the loss of its biomass Lane et al. (2016) and about half of its soil C pool DeLaune and White (2011). When the hydrological alteration disconnects the coastal wetland from the tidal influence, an additional large increase in GHG emissions occurs that yields a radiative forcing an order of magnitude larger than that of automobiles (Kroeger et al., in final review). Hydrological alterations that do not result in an immediate wetland loss increase the vulnerability of the coastal wetland to sea level rise Weston, (2014), resulting in an eventual loss of its C pools unless the wetland can move higher in the tidal frame. The conversion of coastal wetlands, on the other hand, entails the quick loss of its soil and biomass C pools, yielding a C footprint that Kauffman et al. (2017) estimated in 1,440 kg CO_2e per kg of beef or 1,603 kg CO_2e per kg of shrimp produced in the converted (or former) mangrove wetland. Shrimp ponds have a short lifetime and, once abandoned, its C stocks are estimated to be only about 11% of that of mangroves (Ahmed and Glaser, 2016). An additional driver of mangrove degradation frequent in developing countries is the collection of fuelwood for livelihood and charcoal production (Dahdouh-Guebas et al., 2000; Polanía et al., 2015), which is rarely gathered sustainably unless countries' legislation preserve the ecosystem and monitor its compliance to avoid emissions leakage (i.e. mangroves are degraded elsewhere, where regulations are inexistent or unenforced).

This description of the drivers of coastal wetland loss denotes that they are mainly anthropogenic Polanía et al. (2015) which, along with the large C sink capacity of these ecosystems (Fig. 1b), demonstrates the appropriateness of developing a framework for climate mitigation policy for coastal wetland ecosystems (Howard et al., 2017). Such frameworks would reinforce the value of these ecosystems and could function as a management tool for the conservation and enhancement of blue carbon stocks to abate climate change (Ahmed and Glaser, 2016; Lovelock et al., 2017). Macreadie et al. (2017) proposed a framework that not only focuses on restoring coastal ecosystems, but also on managers implementing best-management practices to protect these ecosystems at the catchment level (e.g. restoring hydrology), to increase the efficiency of existing coastal wetlands in sequestering C and thus functioning as C offsets in C markets. Aspects of a REDD+-type framework for coastal wetlands such as C leakage still need to be figured out for its implementation to be successful. Recent studies suggest that including blue carbon ecosystems as a whole in the framework (i.e. including seagrass beds adjacent to mangroves and tidal marshes) could significantly reduce the risk of C leakage, as seagrasses intercept the C exported from mangroves (which conforms up to 80% of the seagrass soil C stock; Chen et al. (2017). A blue carbon integrative approach that accounts for the emission removals from both the coastal wetland and the seagrass beds could be a win-win scenario, as seagrass beds are demonstrated powerful C sinks (Duarte et al., 2010; Mcleod et al., 2011; Fourqurean et al., 2012; Pendleton et al., 2012; Lovelock et al., 2017; among others), whose degradation is human-driven as well (Howard et al., 2017).

To date, the practice of freshwater wetland restoration for climate mitigation and adaptation is hindered by their dual functioning of these ecosystems as both sink and source of radiative forcing. However, management of these ecosystem for the enhancement of the multiple services they provide should not be hampered. A successful market valuation of these services is yet to be developed, but it is known to result in wetlands yielding a positive effect on the environment and human well-being (Costanza et al., 1997; Woodward and Wui, 2001; Mitsch et al., 2013; Mitsch et al., 2014b; Mcleod et al., 2011; Pendleton et al., 2012; The White House, 2011), fostering the restoration practice.

6. Summary and future directions

Carbon sequestration is an important ecosystem service that wetlands provide. The biogeochemistry underlying the process of C sequestration in wetlands is relatively well understood. Wetland's recent inclusion in international policy frameworks as part of climate change adaptation schemes and private initiatives to trade ecosystem services can constitute a significant incentive to manage wetlands sustainably worldwide. This motivation, along with the definition of standard methodologies to facilitate accounting for this service in net GHG emissions assessments, should set the ground for practitioners that manage carbon balanced landscapes, to include wetland creation, restoration, and conservation in their portfolio.

The compilation and analysis of the data gathered from 110 peer-reviewed publications suggest a Shelford-type relationship between C sequestration rates and soil accretion rates for the most studied wetland ecosystems under the Ramsar Classification System. This relationship is explained by hydrogeomorphic settings, unique and characteristic of each wetland class. It further suggests higher sequestration at intermediate soil accretion rates, and highlights the importance of accurate measurement of soil accretion rates when determining C sequestration in wetlands. Future frameworks accounting for ecosystem services that include C sequestrations should therefore address specific methods to measure accretion rates and establish standard methodologies.

Wetlands in tropical climate regions exhibit the highest C sequestration rates, followed by wetlands in temperate and boreal climates. This finding diverges with the number measurements conducted in each climate, since wetlands in boreal and temperate climates have been the most studied to date. The relatively scarce knowledge on tropical wetland biogeochemistry, especially on tropical freshwater wetlands, results in a research gap regarding factors affecting SOM decomposition and mineralization these ecosystems.

In the last section of this manuscript we described key considerations regarding the use of the GWP in current GHG accounting guidelines (and thus C sequestration). We reckon that the inclusion of these considerations in future editions of GHG accounting guidelines could foster the inclusion of C sequestration in (some) wetlands in C and ecosystem services trading schemes. To the date, coastal wetlands and their "blue carbon" stock are the ones with the highest potential to be included in C and ecosystem services trading initiatives.

Acknowledgments

This work was possible thanks to a grant from COLCIENCIAS, Colombia (project code: 127565944348). Publication was assisted by National Science Foundation grant number 1619948 with project title: *U.S. Science International Collaborations and Contributions to EcoSummit 2016 "Sustainability: Engineering Change"*. We thank the EcoSummit 2016 Committee and this journal for the elaboration of this Special Issue, and the colleagues that provided help and advice in the elaboration of this study.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecoleng.2017.06.037.

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